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Family Planning and Deforestation: Evidence from the Ecuadorian Amazon

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Abstract

Despite an abundant body of literature exploring the relationship between population growth and forest cover change, comparatively little research has explored the forest cover impacts of family planning use, which is a key determinant of the rate of population growth in many developing country contexts. Using data from a farm-level panel survey in the Northern Ecuadorian Amazon, this paper addresses whether family planning use impacts forest cover change. Longitudinal model results show that after controlling for household life cycle and land use variables, family planning use did not have an independent effect on deforestation, reforestation, or net forest loss between 1990 and 2008. Forest cover change patterns appear indicative of farm life cycle effects. However, family planning use is associated with reduced subsequent fertility among households, suggesting that the relationship between population growth from births and forest cover change may be limited in this setting.

Introduction

Around the world, tropical deforestation is a multicausal and multiscale process, driven by a variety of factors that can act synergistically, though often centered around agricultural expansion (Geist and Lambin 2002). While population factors (such as growth, size, and density) are often included as covariates in models of deforestation, demographic variables are typically treated as controls, and rarely as the causal variables of interest (Ferretti-Gallon and Busch 2014). Indicators of population are often closely correlated with other deforestation drivers, such as economic activity, land quality, and government investment or development programs, creating an endogeneity problem that has made testing the effects of population factors challenging (Angelsen and Kaimowitz 1999). Thus, several reviews have deemphasized the importance of population as a causal factor in deforestation processes because of these endogeneity challenges (Angelsen and Kaimowitz 1999; Geist and Lambin 2002; Lambin et al. 2001).

However, lingering questions remain about the unique role of population growth as a driver of forest loss, given that the demographic trends occurring throughout fragile ecosystems are unlike those found in other rural areas or in cities. Forest loss often takes place around the edges of forests—frontiers—which are home to settler and indigenous populations carving

out subsistence and market livelihoods with small-scale agriculture. Biodiversity-rich forest frontiers are experiencing higher rates of population growth than other rural areas (Williams 2013), and even small changes in population growth near forest frontiers can contribute to large losses of forest cover (López-Carr and Burgdorfer 2013). This rapid population growth is a function of both in-migration (Carr 2009), as well as high fertility rates (Bremner et al. 2009; Carr 2005; Davis et al. 2015), suggesting that these regions may be among the last places to undergo a demographic transition to lower fertility.

Persistently high fertility in forest frontiers is occurring despite a drop in fertility towards replacement rates throughout much of the developing world. Much of this drop can be attributed to the prevalence of family planning programs (Bongaarts 1997). Given the high fertility rates that often exist in forest frontiers, it is clear that many women in these places either are unable to access family planning or opt not to use it because they desire large families. Although a growing literature has developed on the secondary positive impacts of family planning use, such as on poverty reduction or maternal health (Cleland et al. 2006), relatively little research has explored the impacts that family planning use may have on environmental conditions (Engelman 2016).

This paper is designed to fill a gap between the deforestation and family planning literatures, namely, whether a relationship exists between family planning use and forest cover change. This study explores the impact of family planning use on forest cover change in a dynamic and rapidly-changing forest frontier, the Northern Ecuadorian Amazon (NEA), alongside other drivers of fertility and land use decisions. The association between family planning use and fertility change in the NEA is tested, as well as whether effects on fertility from family planning were associated with forest cover change.

Theoretical Context: Population Growth, Family Planning, and Deforestation

One of the most widely debated drivers of deforestation has been the role of population dynamics, and most prominently, population growth. Historically, the debate concerning the role of population growth has centered on neo-Malthusian versus Boserupian explanations for change. The former set of narratives, based on Malthus's arguments that population growth would outstrip natural resource capacity barring changes in technology and inputs, argue that increases in human demand for agricultural land due to population growth would lead to forest clearing as increasingly marginal lands are brought into production (Malthus 1993). Boserup (1965) developed the concept of induced agricultural intensification, theorizing that in certain circumstances, population growth would incentivize the use of existing strategies known to improve the efficiency of resource use. The pressures societies face from resource constraints may also spur the development of technological and organizational solutions that support the intensification of resource use (Boserup 1981). Thus, Boserupian narratives do not necessarily see population growth as harmful for forest cover due to the behavior changes that scarcity incentivizes.

Carr et al. (2005) outline some of the key challenges of researching the relationship between population growth and deforestation, most notably the spatial variability in examining

deforestation drivers and outcomes, as well as the timing of when population growth occurs, which can impact whether neo-Malthusian or Boserupian relationships between population growth and forest cover predominate. Altering the scale of these variables in models can change results dramatically. Timing matters because population growth at different stages of development can have varying effects on forest cover. Issues of scale and timing are discussed in turn.

Globally, researchers have sought to understand the relationships between population growth and forest cover change at different scales. Early cross-national research exploring the relationship between population growth and deforestation examined forest cover and national-level population growth. This work was able to test whether nationwide population growth created pressure for deforestation, and in numerous cases, found a positive relationship between population growth and deforestation, supporting a neo-Malthusian narrative (Bhattarai and Hammig 2001; Ehrhardt-Martinez 1998; Rudel 1989). However, this research left unanswered the question of whether rural population growth is a more significant determinant than urban growth. Research based on data from the 1980s and 1990s suggests that urban population growth has a negative relationship with deforestation (Jorgenson and Burns 2007). Wright and Muller-Landau (2006) argue that because rural population density has been shown to be negatively correlated with forest cover in many circumstances, that rapidly growing urban populations will reduce pressure on forests. However, more recent research has found the opposite, that deforestation may be more related to urban growth than rural growth due to higher consumption levels in cities (DeFries et al. 2010). Part of the explanation of this divergence may be that more recent processes of globalization and opportunities for agricultural intensification are altering previous patterns of population-deforestation interactions. Global trade is serving as a mechanism that allows population growth and resource harvesting to act as spatially segregated processes (Lambin and Meyfroidt 2011; Meyfroidt et al. 2010), with land clearing in developing regions often driven by rising demand and affluence from the Global North (Weinzettel et al. 2013).

In the context of South America, population-deforestation relationships have varied across scales, with high rates of deforestation sometimes occurring despite relatively low rates of rural population growth, and even population decline in some regions (Carr et al. 2009; Perz 2002). This suggests that the patterns of deforestation in the region are increasingly capital-led and mechanized in part because of the growing road infrastructure in the region that facilitates access to forests as well as transport of resources to markets outside of forest frontiers (Barber et al. 2014; Southworth et al. 2011). Moreover, the decline in the need for on-farm labor to clear forests has enabled high rates of rural-urban migration in Latin America without losses in agricultural productivity (Aide and Grau 2004). In addition, a shift to less labor-intensive (though often more capital-intensive) land use patterns, such as cattle ranching, makes it feasible for farms to operate without high labor inputs (Pan et al. 2007; Rudel et al. 2002). Thus, these trends point to the role of urban population growth and demand from urban areas as being an increasingly important driver of forest cover loss in the region. As recent population growth estimates suggest that rural populations in South America are likely to either shrink or grow far more slowly than urban populations in the coming decades (United Nations 2015), urban growth is likely to play an even greater role in impacting future forest cover.

Moreover, the timing of forest settlement may also serve as a key link between population and deforestation and affect whether neo-Malthusian or Boserupian patterns predominate. Pfaff (1999) notes that the impact of population growth on forests may be disproportionately greater for the first settlers to a forest frontier, who clear relatively large forest areas to begin farming, but subsequent population growth results in less clearing (Laurance et al. 2002; López-Carr 2012). However, more recent in-migrants may settle on land more distant from existing settlements, where land is more abundant and more forest is likely to exist. This may lead to higher rates of deforestation among new settlers, even if these individuals were not part of early settlement waves (Brondizio et al. 2009). Forest clearing and changes in land use patterns to more extensive forms such as pasture may also trigger rural-rural migration as some household members are no longer needed for farm labor, which can perpetuate deforestation processes in other parts of the frontier long after initial settlement (Barbieri and Carr 2005).

In addition to literature exploring how scale and timing impact forest cover change, related research has explored how fertility and land use behaviors are interconnected at the household and farm levels, and often express bidirectional relationships. A robust body of literature explores the impacts of land use characteristics on fertility, often through one of two hypotheses, the land-labor demand hypothesis, which specifies that an increase in available land area but limited capital and labor inputs tends to lead to higher fertility in order to provide more labor for farm activities, provided that returns exist for increased labor input (Cain 1985; Caldwell 1978; Stokes et al. 1986), as well as the land security hypothesis, which argues that tenure security can serve as a substitute for childbearing, since secure landholdings can provide households with greater income security, obviating the need to have additional children to provide support in old age (Schutjer et al. 1983). These hypotheses have been tested empirically in a number of settings around the world (Ghimire and Hoelter 2007; Hiday 1978; Pan and López-Carr 2016; Sutherland et al. 2004), and are explored briefly below in the model examining the impact of family planning use on fertility.

The primary aim of this study, however, is to test relationships between fertility decisions (as understood through decisions regarding family planning use) and forest cover change. To explain demographic-land use relationships, much of the existing literature has centered on household life cycle hypotheses. Originating with Chayanov (1986), household life cycle hypotheses argue that land cover change is a function of household size, with the growth of labor in the household (typically young men), likely to be associated with land clearing and land use intensification in order to provide for household consumptive purposes. Various studies exploring deforestation in South America have modeled the impact of household life cycle factors, though the results have been mixed in support of household life cycle theories (Barbieri et al. 2005; McCracken et al. 2002; Perz 2001; Perz and Walker 2002; Pichon 1997; VanWey et al. 2007; Walker et al. 2002). As VanWey et al. (2007) note, one of the key limitations of household life cycle analyses is that they fail to account for non-household factors in determining household labor allocations. Increased trade of crops and livestock leads to production not intended for household consumption, resulting in land use and labor allocation decisions based on factors other than the household population. In addition, vibrant labor markets in many frontier settings mean that households may opt to expand production by hiring non-household members, or participate in these labor markets by

sending household members to work off farm (Barbieri and Pan 2013). This is in line with findings above, that forest cover change is occurring in South America often without additional labor inputs.

An alternative explanation for effects predicted by household life cycle hypotheses is that smallholder farms have their own life cycles, independent of demographic changes, such as through processes of agricultural intensification (Boserup 1965) or extensification (Bilsborrow 1987). Farm life cycle theories posit that farmers tend to move from intensive to extensive agricultural practices on a given piece of land (thus, affecting the amount of forest cover remaining), but that the speed and extent to which this transition occurs depends on the duration of occupancy as well as property size (Aldrich et al. 2012; Barbieri et al. 2005; VanWey et al. 2007). Both household and farm life cycle theories are tested in the models below linking family planning use to forest cover change.

In addition to arguments on how and why population growth impacts forest cover, another body of literature examines the effects of family planning programs and their impact on demographic change. Voluntary, rights-based interventions that promote the use of family planning have been demonstrated to have an impact on lowering fertility rates (Angeles et al. 2005; Pörtner et al. 2011; Sinha 2005). However, despite the enormous reductions in global fertility that have taken place during the past half-century, there are a number of places around the world where transitions to lower fertility have slowed, most notably in rural sub-Saharan Africa (Bongaarts 2008), but also in places like the Brazilian Amazon, where fertility rates continue to be substantially higher than in other parts of the country (Potter et al. 2010). Because of the association that some scholars have made between rural population growth and forest cover loss, particularly in areas where forest clearing and agricultural processes are labor-driven, there is growing pressure to adopt more effective family planning programs (Cleland et al. 2006).

This study examines the role that slower rural population growth, achieved through family planning use, may have on forest cover change. Thus, the study does not address the increasing spatial segregation of population growth and forest cover change that is being observed in some South American forest frontiers. However, the first part of the study covers the 1990s, a period when road access to outside markets from the study site was more limited, and when capital-intensive pasture was less common (Pan et al. 2004). This suggests that there may have been a stronger relationship during this period between rural population growth and forest cover change than would be found today, because farm labor allocation and household consumption may have been more closely related at that time. In addition, this study addresses the potential temporal relationship between population growth and forest cover change because of its focus on early settlers. However, unlike earlier work that centers on the impact of in-migrants, this study emphasizes the role that natural population growth may have in affecting forest cover change.

Context: The Northern Ecuadorian Amazon

This study is set in the Northern Ecuadorian Amazon (NEA), one of the world's foremost biodiversity hotspots, and home to a variety of endemic rainforest species (Bass et al. 2010;

Mosandl et al. 2008). Before the discovery of oil in 1967, the region was largely uninhabited, aside from small groups of indigenous persons (Bilsborrow et al. 2004). Since the discovery of oil, the region has received rapid inflows of in-migrants and financial capital to provide support for the petroleum industry (Gray et al. 2008). Roads installed to lay pipelines for petroleum extraction have also facilitated human settlement deep into the Amazon, opening up virgin rainforest for agriculture, resulting in high levels of deforestation (Finer et al. 2008). In addition, the Ecuadorian government has provided land titles to many migrant farmers, which has formalized migrant colonization and stimulated deforestation processes, as land titles may serve as collateral for agricultural loans (Bilsborrow et al. 2004). The pace of deforestation in the region has slowed since the 1990s, however (Holland et al. 2014). As the frontier has matured, reforestation is also occurring, though primary forest continues to be lost and degraded.

Population growth in the three provinces that make up the NEA (Napo, Orellana, and Sucumbios) has been quite high, fueled by elevated fertility and in-migration from other regions of Ecuador. Between 1990 and 2010, the population in the NEA grew by roughly 131% to over 416,000 persons, compared with 50% population growth nationally during the same period (INEC 2011). However, fertility rates have fallen considerably in the region among settlers (Pan and López-Carr 2016), though very little among indigenous groups (Davis et al. 2015). As the forest frontier has receded, much of this population growth is occurring around the burgeoning urban areas of Coca and Lago Agrio, leading to increasingly large, settled, and concentrated populations in the region (Bilsborrow et al. 2004).

Historically, family planning services in the Amazon have been provided by a mix of public and private sources. The first high-quality survey providing information on family planning access in the Amazon (not restricted to solely the NEA), conducted in 1999, notes that private sources of family planning access, such as pharmacies, tend to be more popular for most contraceptive methods, with the exception of female sterilization (CEPAR and CDC 1999). Although public facilities were required by law to provide modern contraceptives during the survey period, access to such services varied widely within the region, with more accessible and reliable services often found in or near cities, but less so in some rural areas due to shortcomings with commodity distribution and financing (Goicolea 2001; Goicolea et al. 2008).

Regarding land use change, both intensification and extensification of land use have been observed in the NEA (Barbieri et al. 2005; Pan and Bilsborrow 2005). Plot fragmentation has occurred, which in turn incentivizes land uses that require less area, such as annual and perennial crop agriculture, with crops such as coffee and cacao being popular among farmers. Road infrastructure has been expanded, allowing farmers to access increasingly remote lands. This has helped to drive extensification in the region, largely for cattle ranching (Barbieri et al. 2005; Mena et al. 2006). The development of infrastructure has helped to further integrate the NEA into national and global economies, which may affect forest clearing decisions. For instance, recent evidence suggests that most of the timber produced in the NEA is consumed outside of the region (Carrasco et al. 2014). Small-scale farmers, rather than large business interests, have been the primary agents of forest cover

change in the NEA (Pan and Bilsborrow 2005). Previous literature examining forest cover change in the NEA has explored a variety of household- and farm-level indicators linked to deforestation. These include farm size, distance to nearest road, distance to nearest community, male and female head of household level of education, household size and composition, possession of secure title, number of livestock, access to electricity, topography, soil quality, amount of hired labor, time spent by household members in off-farm labor, and number of household outmigrants (Mena et al. 2006; Pan et al. 2007; Pan and Bilsborrow 2005), factors which are accounted for in the analysis below.

Methods

This study combines the use of spatial and household survey data. Spatial data representing forest cover were extracted from a dataset produced by Ecuador's environment ministry (Ministerio del Ambiente; MAE) which provides information on forest and non-forest cover in 1990, 2000, and 2008 (MAE 2011). Forest cover was defined using IPCC guidelines, with forest defined as tracts at least 1 ha in size with trees of at least 5 m in height, and with a minimum of 30% canopy cover, which includes primary forest, as well as some areas of secondary forest (MAE 2012). Deforestation and reforestation data using farm boundaries (as described below) were extracted separately, and data on net change in forest cover were calculated. MAE data were generated using a combination of Landsat and ASTER satellite imagery with 30 m² resolution, and ground-truthed using over 3,500 calibration points (MAE 2012). Pixel-level changes in forest cover are used to determine deforestation and reforestation over time. These data were chosen over other recent forest cover change time series (e.g., Hansen et al. 2013), because they provide a complete set of validated forest cover data for the time series spanning both of the 1990s household surveys and beyond.

In 1990 and 1999, a UNC-led team conducted a household panel survey in the NEA, exploring the impact of settlers on demographic and land use change in the region (Bilsborrow et al. 2004). Data from the survey have been used to explore a range of demographic and economic topics such as fertility (Carr et al. 2006; Pan and López-Carr 2016), migration (Barbieri and Carr 2005; Barbieri et al. 2009), off-farm employment (Barbieri and Pan 2013), and deforestation and land use change (Barbieri et al. 2005; Mena et al. 2006; Pan et al. 2004, 2007; Pan and Bilsborrow 2005; Pichon 1997).

The UNC team surveyed households on farms of approximately 50 ha (*finca madres*) designated in the NEA, and is longitudinal with respect to the farms, not to the households (Barbieri et al. 2005).¹ Farms were sampled using a two-stage process, first by selecting settlement areas (sectors), and then within each sector, selecting a sample of farms. Both of these steps were conducted using probability proportional to size sampling (Bilsborrow et al. 2004). While most of the farms in 1990 contained only one household, a process of subdivision occurred during the study period, resulting in substantial growth in the number of households on farms by 1999. The 1990 study contained 418 households on 398 farms, while the 1999 study surveyed most of the previous households, as well as new households that had developed on the original farms, yielding a total of 653 households on 377 farms with complete questionnaires. The 1990 sample represented roughly 6% of the region's population at the time (Bilsborrow et al. 2004). Not all of the original 1990 households were

represented in the 1999 study, as some households left the region entirely and a small number of farms were not resampled due to security concerns. The timing of the survey coincides well with the collection of spatial data, making it ideal for use with the MAE imagery. Imagery from 1990 is paired with 1990 survey data, and 2000 imagery is paired with 1999 survey data in the forest cover change models below.

Separate questionnaires were implemented for male and female heads of household, with the former asked primarily about land use, livestock, and labor practices while the latter were asked about household demographics (including fertility and migration), family planning use, and living conditions. Male interviewers conducted interviews with male household heads and female interviewers with female household heads. Additionally, fieldwork by UNC staff established farm-level boundaries for 367 farms. All farms in six sectors did not have boundaries established due to data gathering limitations in the field, and researchers were unable to establish precise boundaries for each household plot within the farms. Each household surveyed had a minimum of 1 ha of land in its possession (households with less land were excluded from data collection on land use practices).

Data for the models below comes from both the male and female household surveys. However, only female heads of household of reproductive age (15–49) were asked whether they currently use family planning. Other female members of the household, even if they were of reproductive age, were not asked about family planning use. Each household has only one female head. Thus, the models only include family planning information for one female per household. Family planning use was recorded as a categorical variable, with households classified into one of three categories. Households with a female head age 15–49 were classified as users or nonusers, while households with a female head outside of that age range, with an infecund head, or without a female head were classified as ineligible. The method of family planning was also recorded.² Chi-square analyses show that the likelihood of using family planning and the methods chosen did not significantly differ between women in different provinces, suggesting that selective access to certain methods is not affecting the method mix chosen.³

Control variables include household and farm demographic and land use characteristics. Most of these were taken from the household survey, with the exceptions of distance to nearest road, which was generated using road network data from 1990 and 1999 collected by UNC researchers, as well as farm area and distance to nearest community, which were calculated using UNC-collected spatial data on farm boundaries and community locations.

Because precise boundaries for land managed by individual households are not available, forest cover change can only be measured at the farm level. As a result, information from the individual household surveys must be weighted up to the farm level in order to provide their representation at the farm level, since multiple households exist on some farms. Some variables, such as household population, can be added together to yield their farm level equivalents. However, other variables, including family planning use, are not additive, and required the use of weights to yield a farm-level value.⁴

As there has been little research on the linkages between family planning use and environmental change, there are no *a priori* weights to use in aggregating household-level data to the farm level. Models were run with identical independent covariates weighted using three different estimators representing plausible mechanisms for how household-level factors may affect forest cover change. Households on the farm with more land under their control may be more influential in spurring farm-level deforestation as they have more potential land to clear. Therefore, the first weight tested is plot size as a share of farm size. Alternatively, households with a greater share of the farm's forest cover may be more influential in generating deforestation. Thus, the second weight tested uses forest cover on household plots as a share of the overall forest cover on the farm. Finally, existing land use may have no bearing on future deforestation by households within the farm. Rather, each household may opt to deforest based on its consumptive needs on the farm, regardless of land or forest availability. Consequently, the third weight uses number of plots on the farm, and is calculated as $(1/p)$ where p represents the total number of plots on the farm. After running the models below, family planning results did not differ significantly between the three weights, and for simplicity, only results using the plot size weight are shown.

The models below explore the impact of family planning decisions, among other factors, on forest cover change. Three sets of models were run, examining deforestation, reforestation, and net forest loss, using predictor variables from 1990 to calculate forest cover changes between 1990–2000 and predictor variables from 1999 to calculate forest cover changes between 2000–2008. Aggregate pixel counts for forest cover change are used because this indicator more directly examines regional-scale forest change as opposed to an indicator of share of forest cover on the farm, which captures farm-level change. Descriptions of the outcome and predictor variables are presented in Table 1.

In addition to these covariates, interaction terms between time and family planning use and ineligibility were added to the models to test whether family planning had different impacts over the study period. Since the rate of deforestation in the region was not constant during the study period (with substantially higher rates of forest loss from 1990–2000 than between 2000–2008; Holland et al. 2014), it is possible that family planning had different impacts on forest cover during the early settlement period as opposed to the later period of more mature settlement.

Deforestation and reforestation models were run using a negative binomial regression, as number of forest pixels lost or gained is a count outcome with positive integer values. A negative binomial specification (as opposed to a Poisson model) was preferred due to overdispersion in the outcome variables (variance greater than the mean). Two specifications were run, a pooled version, with all farms in the sample at both time points, and a fixed effects version that controlled for the effect of time-invariant farm characteristics, and was limited to the subsample of farms that had observations in both 1990 and 1999.⁵ The pooled model did not yield significantly different results from hierarchical mixed models with random slopes for sector or farm that were also tested. As noted by Allison (2009), a conditional fixed effects model that controls for observed time invariant characteristics cannot be calculated when data are negatively binomially distributed. As an alternative, an unconditional maximum likelihood model with farm-level fixed effects was used to control

for farm-specific time-invariant characteristics. This model generated a unique coefficient for each farm in the panel. In both the pooled and fixed effects models, standard errors were clustered at the sector level to account for within-sector correlation (Huber and Ronchetti 2009). This model is specified as:

$$\ln(Y_{fst+1}) = \alpha + \beta X_{fst} + u_{fs}$$

Where: Y_{fst+1} represents the number forest pixels measured in farm f in sector s at time $t+1$; α is a common intercept; β represents the effect of X_{fst} , a matrix of farm-level variables at time t ; u_{fs} represents between-sector error. Negative binomial model coefficients can be exponentiated to be represented as incidence rate ratios. These may be interpreted as the multiplicative change in event frequency (in this case, deforesting or reforesting one pixel in the time period examined) associated with a one unit change in a model variable, holding all other variables constant. Additionally, a linear regression model was run using random and fixed effects estimators to measure net forest loss. A change in estimator was necessary because net forest loss can have positive or negative values, which cannot be modeled with a negative binomial estimator. Note that in the pooled models, farms with 100% forest cover were excluded from reforestation analyses and farms with 0% forest cover were excluded from deforestation analyses, as such farms are not informative for understanding reforestation or deforestation respectively. Farms that exhibited these characteristics in 1990 were dropped from the fixed effects models. No farms were excluded from the net forest loss analyses because of existing forest cover (or lack thereof).

The results in the models described above are compared against a model that examines the impact of family planning use in 1990 on the number of children living in the household born between 1990 and 1999. If family planning use is predictive of fertility, but family planning use is not predictive of deforestation, then this suggests that the relationship between fertility and deforestation in this context is weak. However, this prediction is limited by the fact that the models are being run at different levels (family planning on fertility at the household level, but family planning on deforestation at the farm level), as well as the fact that the family planning-fertility relationship is only examined in one period of the study (due to lack of a follow-up survey in 2008).

The fertility model was run with the subset of households with female heads of household age 15–40 in 1990 who were the same female heads surveyed in 1999. This age range was chosen to solely examine women who would not have completed their reproductive lifespan by 1999 and measure the effect of family planning use on childbearing as opposed to the effects of advanced age on childbearing. Women who were ineligible to use family planning due to age were excluded from this model. As was the case for the deforestation and reforestation models, due to overdispersion in the number of children living in the household, a negative binomial specification was adopted.

The number of children ages nine or below in 1999 (chosen because of the interval between the two household surveys) were regressed on family planning use and the model covariates discussed above to assess the impact of family planning on fertility. These covariates were chosen to match the covariates above in order to demonstrate the differences between the

drivers of fertility and forest cover change processes in the region.⁶ Variables dependent on spatial data (distance to road and fraction of land in forest) were removed as some of the households lacked this information due to the spatial data limitations discussed above. Internal family units were used to separate children of the female head of household from other children living in the household (such as grandchildren) in order to specifically estimate the impact of the female head's family planning use on fertility.⁷ Standard errors were clustered at the sector level as described above.

Results and Discussion

Deforestation occurred rapidly during the 1990–2000 period, but slowed significantly between 2000 and 2008 as the frontier matured and less forest was available for clearing (Table 2). Reforestation exhibited the opposite pattern, with substantially greater reforestation occurring in 2000 to 2008 period, in line with the reduction in deforestation. As a result, net forest loss was substantial between 1990 and 2000, with 87% of farms showing a net loss of forest during this period. This compares to a negligible decrease in forest cover between 2000 and 2008, suggesting that in the aggregate, reforestation from secondary growth is occurring at roughly the same pace as new deforestation in the study farms. Between 2000 and 2008, 41% of farms experienced forest cover gain, and an additional 13% experienced no net change in forest cover. For all three outcomes, the relatively high standard deviations suggest considerable variability among farms in terms of forest cover changes.

Overall, family planning use in 1990 was negatively and significantly associated with fertility (Table 3). Unsurprisingly, younger women were more likely to give birth, as were women in larger households. Higher educational attainment was associated with fewer births, consistent with other literature (e.g. Derose and Ezech 2005; Lam and Duryea 1999), though in this model, the effect was only significant for the male head. This may be the result of a cohort effect among women in the sample, as those with more education tended to be younger, and thus, had not completed childbearing (Pan and López-Carr 2016). Land use (including plot area), and land tenure variables were not significant in the model, suggesting weak support for the land-labor and land security hypotheses in these data after controlling for household demographics and family planning use. Given that family planning use may be a proxy for fertility intentions (with users less likely to desire additional children), a logistic model examining the relationship between household and land use covariates and family planning use was run. Results from these models (not shown), also showed weak relationships between plot size and family planning use as well as land tenure and family planning use. An important caveat with both sets of models is that the sample size was significantly smaller than the number of households surveyed because many female heads of household changed or were no longer eligible to use family planning between 1990 and 1999.

Despite the strong impact of family planning use on fertility at the household level, no effect was found between family planning use and deforestation, reforestation, or net forest change at the farm level (as measured by changes in forest pixels over time), regardless of the weights tested. Table 4 presents findings with the pooled/random effects models shown in

full, and the fixed effects results for family planning variables shown below as a check on the previous findings. Neither the pooled/RE nor the FE models showed significance for family planning use. This suggests that family planning decisions at the farm level were not independently significant factors in determining forest cover outcomes once other demographic variables were included in the model. Two models are shown for each outcome, with one of the model variants containing time-varying indicators for family planning use and ineligibility. The insignificant results for these covariates illustrate that the impacts of family planning on forest cover change did not significantly change between the two periods. In most cases, control variables had effects in the directions predicted. An alternate model specification that included only farms eligible to use family planning due to the presence of a female head of reproductive age did not significantly change model results and are not shown.⁸

Deforestation model findings (first and second columns of Table 4) suggest that the amount of land deforested is largely a function of the amount of forest available to cut as well as indicators of intensive (crop) agriculture. Although some research in the Amazon has associated male farm labor with deforestation (Marquette 1998; Pan and Bilsborrow 2005), this may not be true in our study because of the evolution of forest cover change patterns since periods of earlier research, as agricultural activities in the region have shifted further towards cattle ranching, which is less labor-intensive, but is strongly associated with deforestation. Such activities often occur later in a farm's existence, which is corroborated by the fact that female head of household age was negatively associated with deforestation. High-quality (black) soil was associated with higher deforestation, suggesting that land more valuable for crop agriculture was more likely to be deforested. Farm area and fraction of farm in forest were both positively associated with deforestation, which is unsurprising given that farms with more trees have more opportunities for deforestation. Distance to nearest road and community were negatively associated with deforestation, consistent with earlier literature on forest cover change in the NEA which suggests that accessibility to forests is strongly associated with whether they will remain standing (Mena et al. 2006; Pan et al. 2004).

Reforestation (third and fourth columns of Table 4) appears to have distinct drivers from deforestation in this context, and is not necessarily a process driven by deforestation drivers moving in the opposite direction. A major driver of reforestation appears to center around cattle ranching and associated farm labor needs. Cattle ranching is land-intensive, but does not require large quantities of labor. This is consistent with the fact that hired labor on the farm was positively associated with reforestation, suggesting that farms which adopted more labor-intensive agricultural strategies had spare land that could be left fallow for secondary reforestation. This fallow land may be later cleared and rotated with cultivated land in line with established patterns of shifting cultivation cycles in the Amazon (Perz and Walker 2002). By contrast, farms that adopted less labor-intensive ranching had less of a need for outside labor. Also consistent with this story is that the number of cattle on the farm was significantly and negatively associated with reforestation, and that less reforestation occurred when owners had land titles (which may incentivize investment in capital-intensive ranching). Additionally, having flat land, which earlier literature notes is often associated with use for pasture (Pan and Bilsborrow 2005), was also negatively associated with

reforestation. Farms where households had been present for longer periods of time experienced more reforestation, possibly because they opted to leave more land fallow as it became exhausted through agricultural activities. In addition, such households may be more likely to have shifted through a farm life cycle, with extensive agricultural practices occurring after earlier intensive practices that may have spurred initial deforestation (Barbieri et al. 2005).

Net forest loss (fifth and sixth columns of Table 4) appears to be largely associated with indicators of cattle ranching, consistent with the explanation for reforestation that suggests farms which were engaged in large-scale cattle ranching were less likely to leave land fallow and allow it to return to secondary forest. Flat land, having more cattle on the farm, and hiring less outside labor are all positively associated with net forest loss. Moreover, farms that held secure land title experienced greater net forest loss. As was true with deforestation, fraction of farm in forest was positively associated with net forest loss, while distance to road and community were negatively associated, suggesting that availability and ease of access to forests is a key driver of their destruction. Thus, unlike deforestation, which does not appear to be proximately associated with indicators of cattle ranching, net forest loss is largely determined by the land use and spatial characteristics that are associated with pasture. In the fixed effects model, family planning ineligibility is significantly related to net forest loss, though this is not informative for understanding the effect of family planning use on forest cover, and is likely associated with a life cycle effect (as farms that are ineligible to use family planning tend to have older heads and are more established than eligible farms).

These findings are interesting in light of the fact that recent research published in this Journal found linkages between fertility and deforestation, though did not test the influence of family planning variables (Pan and López-Carr 2016). As we found a relationship between family planning use and fertility, it would be expected that we would find a relationship between family planning use and deforestation, given findings from previous research. However, the fact that we did not may suggest that differences in study timing (our study periods partially overlap, though we examine a more recent time period, later in the demographic transition in the NEA) and data sources used to measure deforestation (as well as the use of interpolation by Pan and López-Carr) may have led to differences in results. Moreover, our findings also reflect the challenges we faced in aggregating up the family planning variable from the household to the farm level, as the family planning indicator in our study is not reflective of decisions by all women of childbearing age.

Conclusion

These findings suggest that family planning use is connected with fertility in the NEA context, but that family planning use is not associated with forest cover change, even at early stages of settlement. This work suggests that the use of family planning is unlikely to alter deforestation trends in this setting relative to other household- and farm-level drivers of forest cover change. Drivers of fertility center on household demographic characteristics, whereas deforestation drivers tend to be variables associated with families cutting down forest to engage in intensive agriculture. As farms move through a farm life cycle, where owners often elect to replace crop agriculture with higher-value cattle ranching, reforestation

is largely determined by indicators closely associated with cattle adoption. Farmers with large ranches appear reluctant to allow land to become fallow. These patterns are also reflected in the models on net forest change, suggesting that whether land remains in forest depends on the speed and extent to which farmers opt to transition between intensive and extensive agricultural practices. Given the lack of significance of family planning variables in these models, there is a need to examine different pathways between population and land use change, and in particular, to explore how changes in population distributions and consumption patterns are associated with forest cover change.

This analysis is limited by the construction of the family planning variable, which only includes information on the female head at the time of the survey, and lacks information on the regularity of method use, which is indicative of how likely it is that family planning will prevent pregnancy. Additionally, this analysis is unable to solve the endogeneity problem between family planning use and forest cover change. Forest cover may be affected as a result of family planning use, (by reducing population growth, and in turn demand for forest or agricultural products). Conversely, family planning use may be impacted by forest cover (for example, due to increased demand for household labor, women who are engaged in deforestation activities could substitute family planning as a way to provide short-term farm labor instead of spending as much time taking care of children).⁹ In addition, this analysis only addresses on-farm forest cover change, and does not examine family planning impacts on forest cover change on other farms, which may result from the provision of surplus labor due to nonuse of family planning. Additionally, such impacts could occur from rural-rural migration, which is common in the region, particularly among young men who work as farm laborers (Barbieri et al. 2009).

These results suggest the importance of exploring the relationship between family planning and forest cover change at different scales, and in particular examining how family planning use by consumers of products that drive deforestation impacts forest cover change. This work will require linking data on deforestation, sales of commodities from deforested areas, as well as family planning and socioeconomic data from areas where products produced in deforested areas are consumed. Trade flows have already been linked to land cover change and displacement patterns (e.g. DeFries et al. 2010; Weinzettel et al. 2013; Yu et al. 2013), although it should be noted that current methodologies for this research have important limitations associated with them (Kastner et al. 2014). While daunting, such multiscalar analyses, incorporating indicators of consumption as well as demographic and family planning variables are a necessary next step in understanding the relationship between population growth and forest cover change, given the increasingly globalized and marketized drivers of deforestation.

In addition to understanding the ultimate drivers of resource consumption patterns, one of the key challenges for researchers testing hypotheses between family planning use and land cover change is the relative timing of the events that lead to land cover change. Deforestation as a result of human settlement can occur relatively quickly in forest frontiers among initial settlers (Pfaff 1999). However, this deforestation is driven largely by the role of in-migration, not births. As the frontier develops, land is cleared and becomes scarcer as the population grows. It is only then that births play a considerable role in household land use

decisions. Family planning may serve to mediate the relationship between births and land use change, but this takes place well after much of the primary forest cover has been lost. Yet frontier development occurs simultaneously with the progression of household life cycles, making it challenging to disentangle the role, if any, that family planning use has in changing land cover with other long-term trends such as household aging and the fragmentation of property among a growing number of households.

Thus, the lack of significance of family planning as a factor influencing proximate forest cover in short time spans suggests that if there are positive impacts on forest cover resulting from family planning use, these are likely to be diffuse and occur over long time frames. The adoption of other policies to fight deforestation, by expanding protected areas and creating payments for ecosystem services programs (as Ecuador has done with the launch of its *Socio Bosque* payments for ecosystem services program; De Koning et al. 2011), are likely to more effectively promote conservation goals than family planning does. However, family planning is likely to improve development and health outcomes for women by reducing unwanted fertility. These benefits, even in the absence of separate impacts on conservation outcomes, suggest that family planning should continue to be promoted in developing country settings.

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Table 1

Variable Descriptions

Dependent Variables	Description
Deforestation	Number of forest pixels changed to nonforest
Reforestation	Number of nonforest pixels changed to forest
Net Forest Loss	Net loss of forest pixels (deforestation-reforestation)
Farm Variables	
Time	Household measured in 1990 (t=0) or 1999 (t=1)
Population*	Number of individuals living on farm
Working-Age Males*	Number of working-age males (age 15–64) living on farm
Outmigrants*	Number of migrants departed from farm (from 1980–1990 for 1990 survey; from 1990–1999 for 1999 survey)
Off-Farm Employment*	Months of off-farm labor provided by individuals on farm in 12 months prior to survey date
Hired Labor*	Months of labor hired by households on farm in 12 months prior to survey date
Cattle*	Number of cattle on farm
Female Head of Household Age*	Mean age of female household heads on farm
Subdivisions	Number of subdivisions on farm
Forest Cover Share	Proportion of farm with forest cover
Farm Area*	Area of farm in hectares
Distance to Road	Walking distance to nearest road
Distance to Community	Euclidean distance to nearest community
Household Variables	
Family Planning Use	Current use of family planning method by female head of household
Family Planning Nonuse	Current nonuse of family planning method by female head of household (reference in models)
Family Planning Ineligibility	Ineligibility for family planning use (due to age, infecundity, or lack of female head)
Years of Education, Male Head	Estimated years of education of male head of household.
Years of Education, Female Head	Estimated years of education of female head of household.
Land Title	Possession of land title in household
Years Present	Number of years household has lived on farm
Flat Land	Plot consists primarily of flat land
Black Soil	Black soil on household's plot
Electricity	Electricity available in household

Note: For use in the fertility model in Table 3, variables with asterisks (*) were not added from different households to yield farm values. Rather, since this model was run at the household level, the values for these variables from each household were used instead. For use in the farm-level models in Table 4, household variables are scaled up to the farm level using household plot-size weights.

Table 2

Variable Summary Statistics

Variables		1990–2000		2000–2008	
Dependent Variables	Unit	Mean	Std. Deviation	Mean	Std. Deviation
Deforestation	pixels	152.66	117.18	35.30	42.59
Reforestation	pixels	8.41	21.47	33.86	49.10
Net Forest Loss	pixels	144.25	122.64	1.44	66.49
Farm Variables		1990		1999	
Time	binary	0	0	1	0
Population	persons	7.61	4.46	10.33	7.74
Working-Age Males	persons	2.16	1.40	3.11	2.40
Outmigrants	persons	0.84	1.29	0.66	0.65
Off-Farm Employment	months	2.73	6.52	0.95	1.22
Hired Labor	months	6.10	12.40	1.70	2.52
Cattle	integer	8.46	25.61	8.36	17.35
Female Head of Household Age	years	40.21	12.33	39.85	10.94
Subdivisions	integer	1.11	0.39	1.78	1.28
Forest Cover Share	fraction	0.61	0.31	0.34	0.27
Farm Area	ha	47.90	11.34	47.94	11.22
Distance to Road	km	0.90	1.10	0.57	0.37
Distance to Community	km	3.59	2.50	3.53	2.42
Household Variables					
Family Planning Use	fraction	0.23	0.41	0.31	0.43
Family Planning Nonuse	fraction	0.43	0.49	0.32	0.43
Family Planning Ineligibility	fraction	0.34	0.47	0.37	0.45
Years of Education, Male Head	years	4.28	2.17	7.62	2.28
Years of Education, Female Head	years	3.33	2.48	6.72	3.01
Land Title	binary	0.57	0.49	0.55	0.47
Years Present	years	9.38	5.04	12.64	8.10
Flat Land	binary	0.41	0.49	0.35	0.46

Variables		1990–2000		2000–2008		
Dependent Variables		Unit	Mean	Std. Deviation	Mean	Std. Deviation
Black Soil		binary	0.65	0.48	0.47	0.48
Electricity		binary	0.15	0.36	0.39	0.47

Sample Size 332 farms (1990); 340 farms (1999)

Note: This table shows means for household variables that have been aggregated to farm values using the household plot size weight.

Table 3

Fertility 1990–1999 Model Results

	Number of Children Born
Family Planning Use	0.642 **
	(0.139)
Female Head of Household Age	0.930 ***
	(0.0126)
Household Size	1.063
	(0.0407)
Working-Age Males	0.783 **
	(0.0909)
Migrants	1.161
	(0.116)
Years of Education, Male Head	0.922 **
	(0.0370)
Years of Education, Female Head	0.982
	(0.0408)
Off-Farm Employment	0.983
	(0.0205)
Hired Labor	0.994
	(0.0157)
Cattle	1.008
	(0.00964)
Land Title	0.760
	(0.166)
Years Present	0.987
	(0.0158)
Black Soil	1.134
	(0.193)
Flat Land	0.862
	(0.181)
Electricity	0.997
	(0.272)
Subdivisions	0.985
	(0.122)
Plot Area	1.003
	(0.00502)
Constant	24.82 ***
	(13.64)

	Number of Children Born
Sample Size (Households)	123

Negative binomial regression model with incidence rate ratios presented. Cluster-robust standard errors are in parentheses below.

*
p<.10;

**
p<.05;

p<.01

Table 4

Forest Cover Change 1990–2008 Model Results

	Deforestation		Reforestation		Net Forest Loss	
Family Planning Use	1.110	1.122	0.872	0.792	1.070	-6.603
	(0.113)	(0.119)	(0.131)	(0.209)	(8.867)	(13.65)
Family Planning Ineligibility	1.071	1.125	1.105	1.235	5.450	5.912
	(0.0991)	(0.121)	(0.223)	(0.359)	(10.64)	(15.37)
Time	0.486***	0.508***	3.310***	3.420***	-72.76***	-77.07***
	(0.0849)	(0.110)	(0.749)	(1.119)	(11.24)	(14.85)
Family Planning Use * Time		0.968		1.157		14.54
		(0.183)		(0.360)		(16.54)
Family Planning Ineligibility * Time		0.902		0.803		0.152
		(0.152)		(0.264)		(17.57)
Female Head of Household Age	0.993**	0.993**	1.001	1.001	-0.482	-0.493
	(0.00356)	(0.00358)	(0.00886)	(0.00871)	(0.487)	(0.491)
Farm Population	1.008	1.008	1.009	1.007	0.855	0.856
	(0.0110)	(0.0107)	(0.0252)	(0.0255)	(1.545)	(1.514)
Working-Age Males	1.024	1.024	0.989	0.982	1.163	0.985
	(0.0289)	(0.0289)	(0.0553)	(0.0538)	(2.404)	(2.406)
Outmigrants	0.982	0.979	0.909	0.903*	-4.961	-4.955
	(0.0371)	(0.0381)	(0.0533)	(0.0541)	(3.379)	(3.606)
Male HoH Years of Education	0.966*	0.966*	1.026	1.027	-1.582	-1.509
	(0.0174)	(0.0176)	(0.0266)	(0.0265)	(1.447)	(1.446)
Female HoH Years of Education	0.983	0.984	0.982	0.983	-1.324	-1.374
	(0.0139)	(0.0142)	(0.0216)	(0.0210)	(1.192)	(1.147)
Off-Farm Employment	1.000	0.999	0.980	0.980	-0.167	-0.0735
	(0.00502)	(0.00466)	(0.0205)	(0.0206)	(0.582)	(0.583)
Hired Labor	0.998	0.998	1.022**	1.021**	-0.827*	-0.803*

	Deforestation		Reforestation		Net Forest Loss	
	(0.00558)	(0.00569)	(0.0108)	(0.0107)	(0.472)	(0.482)
Number of Cattle	1.002	1.002	0.984***	0.984***	0.418**	0.425**
	(0.00253)	(0.00245)	(0.00344)	(0.00337)	(0.208)	(0.204)
Land Title	1.020	1.015	0.804*	0.788*	20.41**	20.17**
	(0.0679)	(0.0684)	(0.106)	(0.102)	(8.702)	(8.631)
Years Present	1.004	1.004	1.020**	1.021**	0.399	0.403
	(0.00594)	(0.00608)	(0.0103)	(0.0102)	(0.782)	(0.813)
Black Soil	1.216*	1.213*	0.881	0.892	12.79	13.06
	(0.130)	(0.129)	(0.114)	(0.112)	(9.668)	(9.864)
Flat Land	1.091	1.091	0.622**	0.617**	28.42**	27.89**
	(0.109)	(0.110)	(0.125)	(0.126)	(12.54)	(12.49)
Electricity	0.949	0.947	0.954	0.942	6.483	6.935
	(0.116)	(0.118)	(0.178)	(0.172)	(11.02)	(10.84)
Subdivisions on Farm	0.974	0.978	0.966	0.985	-3.080	-2.770
	(0.0646)	(0.0648)	(0.137)	(0.138)	(6.710)	(6.616)
Farm Area	1.017***	1.017***	1.004	1.005	0.796	0.806
	(0.00475)	(0.00465)	(0.00481)	(0.00463)	(0.829)	(0.832)
Fraction of Farm in Forest	7.971***	8.065***	0.431	0.440	277.4***	276.1***
	(2.097)	(2.126)	(0.257)	(0.265)	(22.64)	(23.05)
Distance to Road	0.750***	0.750***	0.835	0.812	-27.73***	-27.86***
	(0.0547)	(0.0539)	(0.247)	(0.246)	(6.087)	(6.162)
Distance to Community	0.952***	0.952***	0.962	0.963	-7.406***	-7.320***
	(0.0179)	(0.0178)	(0.0323)	(0.0325)	(2.666)	(2.689)
Constant	30.72***	30.04***	16.05***	15.22***	-19.07	-17.40
	(10.23)	(10.04)	(12.33)	(11.65)	(47.02)	(47.58)
Sample Size (Farms)	599		639		672	
Fixed Effects Models						
Family Planning Use	1.041	1.023	1.207	1.658	-5.303	-10.47

	Deforestation		Reforestation		Net Forest Loss	
	(0.206)	(0.254)	(0.299)	(0.623)	(9.834)	(14.01)
Family Planning Ineligibility	1.291	1.342	1.087	1.197	14.74 *	18.86 **
	(0.240)	(0.321)	(0.344)	(0.428)	(8.322)	(8.644)
Family Planning Use * Time		1.029		0.531		9.231
		(0.319)		(0.238)		(14.38)
Family Planning Ineligibility * Time		0.936		0.768		-6.673
		(0.277)		(0.352)		(12.49)
Sample Size (Farms)	592		570		628	

Incidence rate ratios presented for deforestation and reforestation negative binomial regression models, regression coefficients presented for net forest loss linear regression models. Cluster-robust standard errors are presented in parentheses below. An indicator of missing female head of household age at the farm level is included in all models but not shown.

* p<.10;
** p<.05;
*** p<.01